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Successions of phytobenthos species in a Mediterranean transitional water system: the importance of long term observations

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Abstract

The availability of quantitative long term datasets on the phytobenthic assemblages of the Mar Piccolo of Taranto (southern Italy, Mediterranean Sea), a lagoon like semi-enclosed coastal basin included in the Italian LTER network, enabled careful analysis of changes occurring in the structure of the community over about thirty years. The total number of taxa differed over the years. Thirteen non-indigenous species in total were found, their number varied over the years, reaching its highest value in 2017. The dominant taxa differed over the years. The number of species in each taxonomic division also varied. In addition to the centuries-old exploitation of its biotic resources, mainly molluscs, the basin has been subject for a long time to a range of anthropogenic driving forces linked to urbanisation, shipbuilding, agriculture and military activities, which have caused chemical and biological pollution, eutrophication and habitat destruction. It may therefore be assumed that these changes were closely related to human activities. Indeed, it was observed that the reduction of only one of these forces, i.e. urban sewage discharge, triggered the recovery of phytobenthos. Therefore, it may be assumed that if the anthropogenic pressure on the Mar Piccolo was eased, it could once again become the paradisiacal place it was held to be in ancient times.

Keywords

LTER, Mar Piccolo, Mediterranean Sea, phytobenthos, transitional water systems

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Introduction

Human beings play a fundamental role in the ongoing degradation of coastal marine ecosystems (Bianchi et al. 2014). Coastal lagoons and transitional water systems (TWSs) generally are frequently chosen for human activities, due to their physico-chemical and biocoenotic features. The main anthropogenic forcing factors are aquaculture, fishing and shipping, as well as tourism and recreation (Newton et al. 2014), with eutrophication and chemical pollution the most important effects on the environment (Pérez-Ruzafa et al. 2011a). Recently, biological pollution, caused by the introduction of non-indigenous species (NISs), has also become a significant issue in TWSs (Lotze et al. 2006). Heavy anthropogenic use severely damages such environments, which in contrast should be preserved by sustainable management. Extensive knowledge of the biotic and abiotic components of these systems is thus necessary, as well as an understanding of the cause-effect relationship between anthropogenic processes and environmental feedback (Pérez-Ruzafa et al. 2011a).

The availability of historic data on benthic populations could allow better measurement of the changes (Bianchi et al. 2014; Husa et al. 2014). Historical datasets could also make it possible to evaluate the effects of environmental recovery measures and to predict possible reactions to future changes (Gosz et al. 2010; Mirtl 2010; Bertoni et al. 2012). To this end, long-term ecological research (LTER), i.e. studies embracing long time-scales, could enhance our understanding of interactions amongst biotic components and the environment and help to distinguish between natural ecological processes and changes due to human actions (Bertoni et al. 2012).

The Mar Piccolo of Taranto (southern Italy) has belonged to the Italian LTER network since 2011 (LTER_EU_IT_095; https://deims.org/ac3f674d-2922-47f6-b1d8-2c91daa81ce1) (Cecere and Petrocelli 2012). It is a semi-enclosed coastal basin with lagoon features, which falls into the category of transitional waters (Viaroli and Basset 2009). Its phytobenthos has been studied since the second decade of the 20th century, albeit not continuously (Cecere and Petrocelli 2009). Comparison of historical floristic data dating back to the 1920s (Pierpaoli 1923; Cecere and Saracino 1999) with data collected in 1986, enabled initial assessment of the changes affecting phytobenthic assemblages in the Mar Piccolo over a period of sixty-four years (Cecere et al. 1991a). Subsequently, some other floristic lists, arising from studies carried out in the basin in 2001 and 2008 with a range of methods, became available (Sassanelli 2011; E. Cecere and A. Petrocelli, unpublished data). In addition, in 1989, a one-year quantitative study was performed, which gave the first ever information on the standing crop of seaweed in the Mar Piccolo and detected the dominant species (Cecere et al. 1992).

The aim of the present study was to increase our knowledge of the structure of phytobenthic assemblages in the Mar Piccolo of Taranto and to reconstruct the history of macrophytic communities in terms of both species composition and dominance changes during the last 30 years. For this purpose, seasonal quantitative data, collected at various sites in the Mar Piccolo in 2008 (Project VECTOR http://vector.conismamibi.it) and from spring 2011 to winter 2018 (I-LTER network), were carefully

analysed and compared with the floristic data collected in 1989. At the same time, again on the basis of already published data, speculative parallels were drawn with the demographic and socio-economic development of the town of Taranto (https://tiny-url.com/yacl8t6t; Nistri 1988).

Methods

Study site

The Mar Piccolo of Taranto is a lagoon-like basin located north of Taranto. It has a surface area of 20.72 km^2 and is divided into two sub-basins known as the First Inlet, to the west and the Second Inlet, to the east (Fig. 1). Hard substrata account for a limited area and are mostly artificial, consisting of concrete wharves and facilities for mussel farming. A small reef and a ruined stone pier, together with a few scattered stones and mussel shells, are the only natural hard substrata. Recent observations both *in situ* and by remote sensing showed that the anthropogenic impact on the Mar Piccolo is very high in the First Inlet but less severe in the Second Inlet and arises mainly from lines, poles and frames used for mussel farming (Bracchi et al. 2016). Hydrodynamism is limited, as is exchange with the adjacent Mar Grande basin (i.e. Mediterranean Sea), which occurs mainly through the Canale Navigabile (Fig. 1). The mean annual seawater surface temperature ranges from 7.5 °C to 32.3 °C and seawater surface salinity from 33.0 to 37.7 ‰ (Cecere et al. 2016a). Those values are strongly affected by inputs of freshwater from submarine springs and a few small rivers (Cecere and Petrocelli 2009).

Field activities

In 2008, sampling exercises were carried out at monthly intervals at three sites, two in the First Inlet (Station A and Station B) and one in the Second Inlet (Station C).

Station A (40°29'35"N, 17°14'17"E) was characterised by soft bottoms and artificial hard substrata, mainly concrete blocks. Station B (40°30'01"N, 17°15'10"E) was characterised by a soft muddy bottom. Station C (40°29'40"N, 17°19'18"E) had a soft muddy bottom with scattered concrete blocks.

In the period spring 2011 - winter 2018, four sites were seasonally sampled, two in the First Inlet (Station 1 and Station 2) and two in the Second Inlet (Station 3 and Station 4) (Fig. 1).

Station 1 (40°28'46"N, 17°13'41"E) was the only urban station, located in the old town. Artificial hard substrata prevailed, composed mainly of discarded plastic used in mussel farming activities but also concrete quays. Station 2 (40°30'03"N, 17°15'30"E) was located next to the mouth of a small river, with a soft muddy bottom and a few artificial substrates, mainly unlawfully built concrete quays used

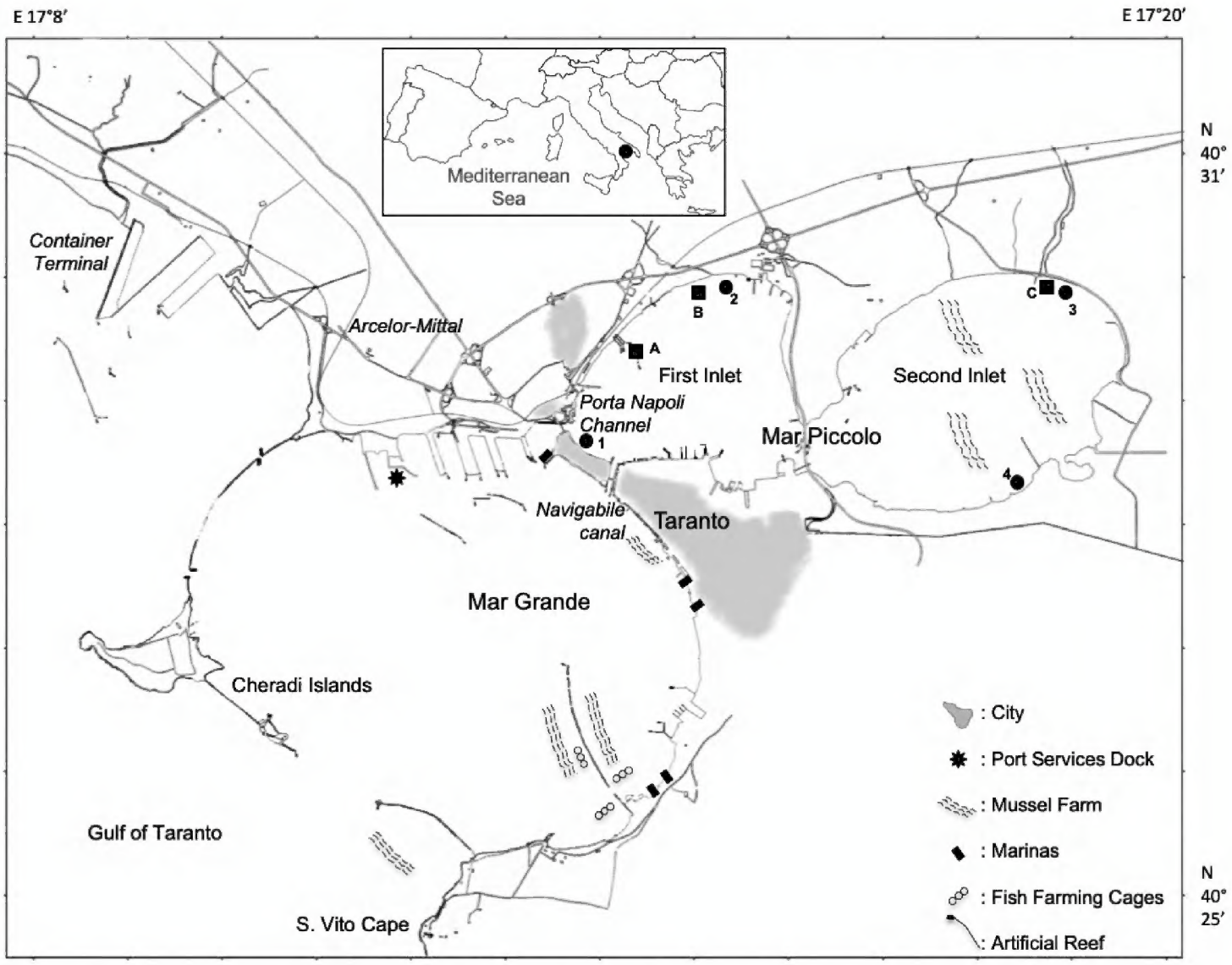

Figure 1. Location of Mar Piccolo in the Gulf of Taranto (Mediterranean Sea). ■ 2008 stations; ● 2011–2018 stations.

by mussel farmers. Station 3 (40°29'39"N, 17°19'22"E) was the only station characterised by natural hard substrata, rare in the Mar Piccolo, i.e. small rocks and the remains of a stone dock. It was close to the mouth of a small river. Station 4 (40°28'20"N, 17°18'25"E) was located in a zone where mussels and other seafood are manually cleaned, bagged and sold. A soft muddy bottom and concrete platforms were the main substrata.

The samples were handpicked from within a 50×50 cm square, randomly placed on the bottom. Three replicates were collected at each station. Seaweed thalli were stored in plastic bags and transported to the laboratory within a few hours. There, the macrophytes were sorted and each species was identified and weighed on a triple-beam balance. Data were expressed as kg wet weight m^{-2} (hereafter kg m^{-2}). A list of species was compiled for each dataset, including the unpublished floristic data from 1989, with nomenclature as in Guiry and Guiry (2018). In order to make the 1989 and 2008 datasets comparable, only data relating to the months that were also sampled in 2011–2018 (i.e. March for winter, May for spring, September for summer, December for autumn) were considered.

Data analysis

The quantitative data recorded in 2008 were not processed, since the sampling sites were not the same as those of 2011–2018. Therefore, only the total number of species, taxonomic divisions, biogeographical elements and dominant species were compared.

Since seasonal samplings started in spring 2011, to allow statistical tests to be performed on all four seasons of the year, the analysis was carried out during a period embracing the spring, summer and autumn of one year together with the winter of the following year (spring 2011 - winter 2012).

All univariate and multivariate analyses were performed using PRIMER v.6 (Primer-E Ltd., Plymouth, UK).

Two different matrices of the means were constructed from the three replicates collected at each station. The first matrix considered all the identified taxa and the second matrix only the NIS taxa. From the absolute abundance matrices (taxa vs. samples), the Bray-Curtis similarity index was calculated after log(x+1) transformation, in order to remove the effects of orders-of-magnitude differences between samples, to normalise the data and to increase the importance of smaller values, such as the mid-range species (Frontalini et al. 2009).

The PRIMER 'DIVERSE' routine was used to calculate the taxonomic richness (S), taxon abundance (n°taxa), Pielou's evenness index (J') and Shannon-Weaver diversity index (H') for each sample.

The statistical significance of spatial and temporal variations in the community structure across the defined factors, "year", "season", "basin" and "station", were tested by one-way analysis of similarities (ANOSIM). A two-way crossed analysis (year x station) was performed to highlight possible differences from year to year. In addition, bi-dimensional representations of the statistical comparisons amongst the samples collected during the seven years at the four stations were performed by means of non-parametric multidimensional scaling (nMDS) with superimposed hierarchical clustering and a cut-off at 80–90% similarity.

The SIMPER routine was used to identify the species that contributed most to dissimilarities amongst sampling sites (one-way procedure) and to explain the changes in terms of biomass or species composition (two-way procedure, years vs. sites).

All the multivariate tests were performed on the Bray-Curtis similarity matrix including all the identified taxa.

Results

The floristic lists of species collected in 1989, 2008 and 2011–2018 are reported in Table 1. The total number of taxa (Fig. 2, Table 1) differed over the years. Only nine taxa were found in all the considered periods (i.e. *Alsidium corallinum, Chaetomorpha linum, Dictyota dichotoma* var. *dichotoma, D. dichotoma* var. *intricata, Gracilaria bursa-pastoris, G. gracilis,*

Table 1. Floristic list (presence/absence data) of seaweeds and phanerogams recorded in the Mar Piccolo in the different years. B=biogeographic element, A=Atlantic, C=Cosmopolitan, CT=CircumTropical, IP=IndoPacific, M=Mediterranean.

	B	1989	2008	2011	2012	2013	2014	2015	2016	2017	2018
Unidentified Bacillariophyta		+			+	+	+				
Unidentified Cyanophyta		+	+								
Rhodophyta											
Acanthophora nayadiformis (Delile) Papenfuss	IP	+	+								
Agardhiella subulata (C. Agardh) Kraft *et* M.J. Wynne	A	+							+	+	
Aglaothamnion tenuissimum (Bonnemaison) Feldmann-Mazoyer	A									+	+
Aglaothamnion tripinnatum (C. Agardh) Feldmann-Mazoyer	A									+	
Alsidium corallinum C. Agardh	A	+	+	+	+	+	+	+	+	+	+
Alsidium helminthochorton (Schwendimann) Kützing	M	+									
Amphiroa beauvoisii J.V. Lamouroux	A			+	+	+	+	+	+	+	+
Antithamnion cruciatum (C. Agardh) Nägeli	A		+		+	+	+		+	+	
Antithamnion tenuissimum (Hauck) Schiffner	M					+					
Callithamnion corymbosum (J.E. Smith) Lyngbye	A	+	+		+			+	+	+	+
Caulacanthus cf. *ustulatus* (Mertens *ex* Turner) Kützing	C				+	+	+	+	+	+	+
Ceramium cimbricum H.E. Petersen f. *flaccidum* (H.E. Petersen) G. Furnari *et* Serio	C		+	+				+		+	
Ceramium comptum Børgesen	A								+	+	
Ceramium diaphanum (Lightfoot) Roth	C								+		
Ceramium incospicuum Zanardini	M		+					+			+
Ceramium siliquosum (Kützing) Maggs *et* Hommersand	C	+		+	+	+	+		+		
Chondracanthus acicularis (Roth) Fredericq	C		+	+	+	+	+	+	+	+	+
Chondracanthus teedei (Mertens *ex* Roth) Kützing	C		+		+	+	+	+	+	+	
Chondria capillaris (Hudson) M.J. Wynne	C	+	+								
Chondria dasyphylla (Woodward) C. Agardh	C		+								
Chylocladia verticillata (Lightfoot) Bliding	A		+	+	+	+	+	+			
Corallina officinalis Linnaeus	C				+	+	+	+		+	
Dasya baillouviana (S.G. Gmelin) Montagne	CT		+								
Dasya ocellata (Grateloup) Harvey	A		+				+		+		
Dasya punicea (Zanardini) Meneghini	A									+	
Dasya rigidula (Kützing) Ardissone	A				+	+					
Dasysiphonia sp							+	+	+		
Ellisolandia elongata (J. Ellis *et* Solander) K.R. Hind *et* G.W. Saunders	A			+	+	+	+	+	+	+	+
Erythrocladia irregularis Rosenvinge	C				+		+				
Gayliella flaccida (Harvey *ex* Kützing) T.O. Cho *et* L.J. McIvor	C		+	+						+	
Gelidiella lubrica (Kützing) Feldmann *et* Hamel	IP		+		+				+		
Gelidium crinale (Hare *ex* Turner) Gaillon	C				+	+	+	+	+	+	+
Gelidium pusillum (Stackhouse) Le Jolis	C		+		+			+			+
Gigartina cf. *pistillata* (S.G. Gmelin) Stackhouse	A								+	+	
Gracilaria bursa-pastoris (S.G. Gmelin) P.C. Silva	C	+	+	+	+	+	+	+	+	+	+
Gracilaria dura (C. Agardh) J. Agardh	A	+		+							
Gracilaria gracilis (Stackhouse) Steentoft, L.M. Irvine *et* Farnham	A	+	+	+	+	+	+	+	+	+	
Gracilaria longa Gargiulo, De Masi *et* Tripodi	M	+	+								
Gracilariopsis longissima (S.G. Gmelin) Steentoft, L.M. Irvine *et* Farnham	C	+	+	+	+	+	+	+	+	+	
Grateloupia cf. *filicina* (J.V. Lamouroux) C. Agardh	C	+		+	+	+	+		+	+	+
Grateloupia minima P.L. Crouan *et* H.M. Crouan	A							+		+	
Grateloupia turuturu Yamada	IP			+	+	+	+	+	+	+	+

	B	1989	2008	2011	2012	2013	2014	2015	2016	2017	2018
Griffithsia schousboei Montagne	A	+									
Herposiphonia secunda (C. Agardh) Ambronn	CT					+		+			
Herposiphonia tenella (C. Agardh) Ambronn	CT	+	+	+		+	+		+		
Heterosiphonia crispella (C. Agardh) M.J. Wynne	A							+			
Huismaniella nigrescens (Feldmann) G. Furnari, Cormaci, Alongi *et* Perrone	M				+						
Huismaniella ramellosa (Kützing) G.H. Boo *et* S.M. Boo	CT		+								
Hydrolithon cruciatum (Bressan) Y.M. Chamberlain	A		+						+		
Hydrolithon farinosum (J.V. Lamouroux) Penrose *et* Y.M. Chamberlain	C			+		+	+		+		
Hypnea cornuta (Kützing) J. Agardh	IP		+	+	+	+	+	+	+	+	
Hypnea musciformis (Wulfen) J.V. Lamouroux	CT		+				+	+	+	+	
Hypnea spinella (C. Agardh) Kützing	CT	+	+				+				
Jania rubens (Linnaeus) J.V. Lamouroux	C						+	+	+	+	
Jania virgata (Zanardini) Montagne	A							+			
Laurencia intricata J.V. Lamouroux	A		+								
Lomentaria clavellosa (Lightfoot *ex* Turner) Gaillon	A				+						
Lomentaria compressa (Kützing) Kylin	M	+									
Lophosiphonia obscura (C. Agardh) Falkenberg	C				+						
Nitophyllum albidum Ardissone	M				+						
Osmundea oederi (Gunnerus) G. Furnari	A		+							+	+
Osmundea pelagosae (Schiffner) K.W. Nam	M									+	
Peyssonnelia bornetii Boudouresque *et* Denizot	M			+							
Phymatolithon calcareum (Pallas) W.H. Adey *et* D.L. McKibbin *ex* Woelkerling *et* L.M. Irvine	A										+
Polysiphonia denudata (Dillwyn) Greville *ex* Harvey	C	+						+			
Polysiphonia elongata (Hudson) Sprengel	A		+						+		
Polysiphonia morrowii Harvey	IP				+		+	+	+	+	+
Polysiphonia subulata (Ducluzeau) Kützing	A	+					+	+		+	
Porphyra linearis Greville	A				+					+	
Porphyra umbilicalis Kützing	C	+				+		+			
Pterocladiella capillacea (S.G. Gmelin) Santelices *et* Hommersand	C			+						+	
Pterocladiella melanoidea (Schousboe *ex* Bornet) Santelices *et* Hommersand	A		+				+		+		
Pyropia leucosticta (Thuret) Neefus *et* J. Brodie	A						+				
Radicilingua reptans (Kylin) Papenfuss	A						+				
Radicilingua thysanorhizans (Holmes) Papenfuss	A	+	+			+	+	+	+	+	+
Rhodymenia ardissonei (Kuntze) Feldmann	A							+	+		
Rhodymenia pseudopalmata (J.V. Lamouroux) P.C. Silva	A	+	+							+	
Rytiphlaea tinctoria (Clemente) C. Agardh	A							+			
Solieria filiformis (Kützing) P.W. Gabrielson	A	+								+	
Spyridia filamentosa (Wulfen) Harvey	C	+	+	+	+	+	+	+	+	+	+
Stylonema alsidii (Zanardini) K.M. Drew	C		+								
Stylonema cornu-cervi Reinsch	A			+							
Wrangelia penicillata (C. Agardh) C. Agardh	CT			+							
Unidentified non geniculate Corallinaceae					+	+					
83		24	34	22	29	25	32	33	34	37	18
Ochrophyta											
Colpomenia peregrina Sauvageau	C				+	+	+	+	+	+	+
Colpomenia sinuosa (Mertens *ex* Roth) Derbès *et* Solier	C							+	+	+	
Cutlera chilosa (Falkenberg) P.C. Silva	M						+				
Cutleria multifida (Turner) Greville	C									+	+
Cystoseira barbata (Stackhouse) C. Agardh	IP	+		+	+	+	+				
Cystoseira compressa (Esper) Gerloff *et* Nizamuddin	A					+	+	+	+	+	

	B	1989	2008	2011	2012	2013	2014	2015	2016	2017	2018
Dictyota dichotoma (Hudson) J.V. Lamouroux v. *dichotoma*	C	+	+	+	+	+	+	+	+	+	
Dictyota dichotoma (Hudson) J.V. Lamouroux *var. intricata* (C. Agardh) Greville	C	+	+	+	+	+	+	+	+	+	+
Ectocarpus siliculosus (Dillwyn) Lyngbye	C	+			+	+	+		+	+	
Feldmannia mitchelliae (Harvey) H.-S. Kim	C								+		
Halopteris filicina (Grateloup) Kützing	C				+					+	
Hincksia dalmatica (Ercegović) Cormaci *et* G. Furnari	M							+		+	+
Nemacystus flexuosus (C. Agardh) Kylin var. *giraudyi* (J. Agardh) De Jong	M						+				
Padina pavonica (Linnaeus) Thivy	CT				+	+	+	+	+	+	+
Petalonia fascia (O.F. Müller) Kuntze	C							+		+	
Scytosiphon lomentaria (Lyngbye) Link	C				+			+	+	+	+
Sphacelaria cirrosa (Roth) C. Agardh	C							+	+	+	
Sphacelaria fusca (Hudson) S.F. Gray	C								+	+	+
Sphacelaria rigidula Kützing	C				+	+					
Undaria pinnatifida (Harvey) Suringar	IP		+								
20		4	3	3	9	8	9	10	11	14	7
Chlorophyta											
Bryopsis corymbosa J. Agardh	A									+	
Bryopsis cupressina J.V. Lamouroux	M								+		+
Bryopsis pennata J.V. Lamouroux	A				+	+	+				
Bryopsis plumosa (Hudson) C. Agardh	C					+	+	+			
Caulerpa cylindracea Sonder	CT		+			+					
Chaetomorpha linum (O.F. Müller) Kützing	C	+	+	+	+	+	+	+	+	+	+
Cladophora dalmatica Kützing	A		+	+							
Cladophora glomerata (Linnaeus) Kützing	C				+	+		+	+		
Cladophora hutchinsiae (Dillwyn) Kützing	C									+	+
Cladophora laetevirens (Dillwyn) Kützing	C		+	+	+	+		+	+	+	
Cladophora lehmanniana (Lindenberg) Kützing	A					+			+		
Cladophora prolifera (Roth) Kützing	A								+		
Cladophora ruchingeri (C. Agardh) Kützing	A	+									
Cladophora rupestris (Linnaeus) Kützing	A		+			+	+				
Cladophora sericea (Hudson) Kützing	C		+						+		
Codium fragile (Suringar) Hariot subsp. *fragile*	IP			+		+	+	+	+		
Ulva compressa Linnaeus	C	+		+				+	+	+	+
Ulva curvata (Kützing) De Toni	A	+	+		+	+	+				
Ulva flexuosa Wulfen	C	+			+	+				+	
Ulva intestinalis Linnaeus	C			+	+	+	+		+	+	+
Ulva laetevirens Areschoug	C	+	+	+	+	+	+	+	+	+	+
Ulva prolifera O.F. Müller	C	+	+		+	+		+	+	+	
Ulva pseudorotundata Cormaci, G. Furnari *et* Alongi	A	+	+	+	+	+	+	+	+		+
Ulva rigida C. Agardh	A	+					+		+	+	+
Umbraulva dangeardii M.J. Wynne *et* G. Furnari	A			+	+	+	+	+	+	+	
Valonia macrophysa Kützing	CT				+						
26		9	10	9	12	16	11	10	15	11	8
Spermatophyta											
Cymodocea nodosa (Ucria) Ascherson	A			+	+	+	+	+	+	+	+
Ruppia cirrhosa (Petagna) Grande	C				+						
Zostera noltei Hornemann	A								+		
3		0	0	1	2	1	1	1	2	1	1
TOTAL											
132		37	47	35	52	50	53	54	62	63	34

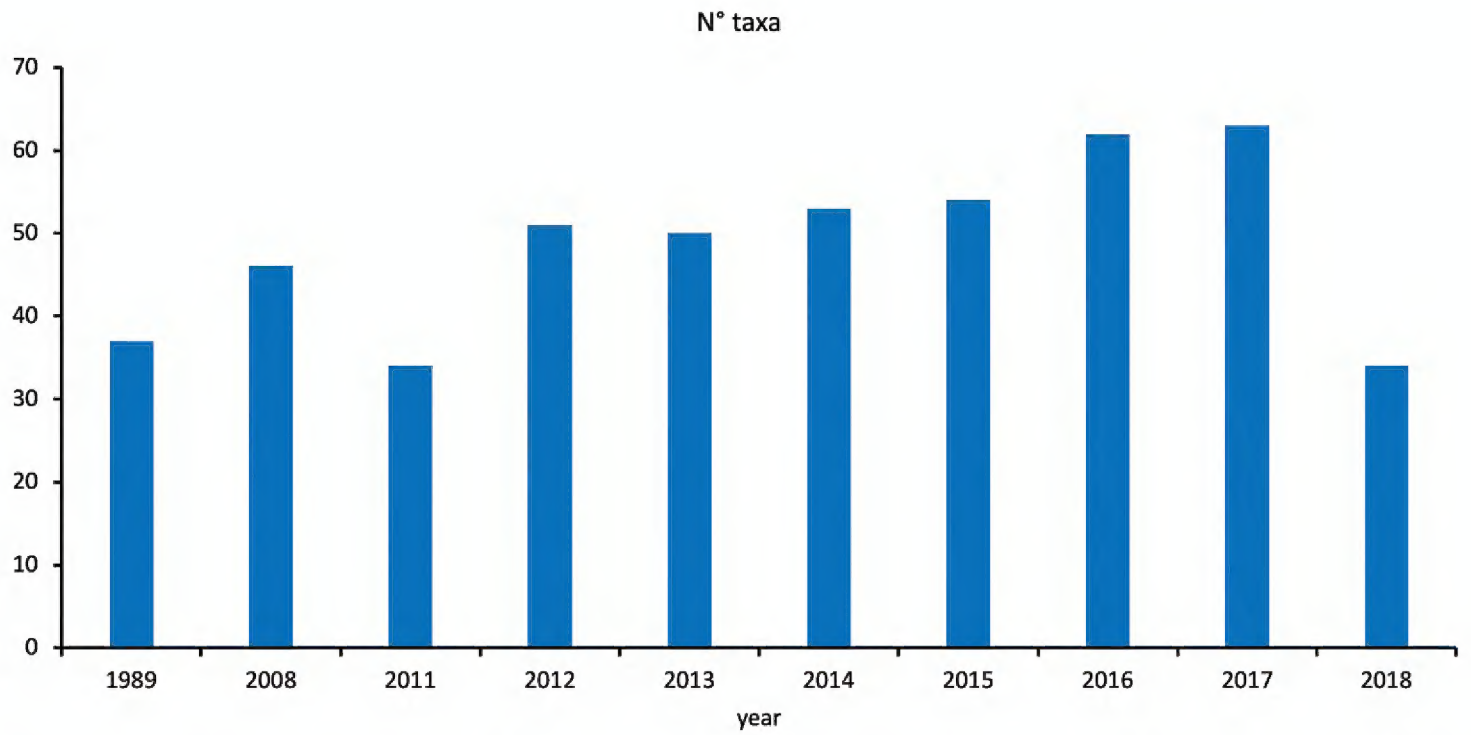

Figure 2. Trend in the total number of species. 2018 data for winter only. In addition to the analysed years 2008 and 2011–2018, unpublished floristic data from 1989 have been included.

Gracilariopsis longissima, Spyridia filamentosa and *Ulva laetevirens*). Seventy-six taxa recorded in 2011–2018 were not found in 1989 and 2008 (Table 1). Three taxa (i.e. *Acanthophora nayadiformis, Chondria capillaris* and *Gracilaria longa*) and unidentified Cyanophyta were recorded in 1989 and 2008 but not found in 2011–2018 (Table 1). Thirteen NISs in total were found. Their number varied over the years, reaching its highest value in 2017 (Table 2).

The number of species in each taxonomic division (i.e. Rhodophyta, Ochrophyta, Chlorophyta and Spermatophyta) also varied (Fig. 3). Rhodophyta were the most numerous in all years. Chlorophyta exceeded Ochrophyta until 2016. Spermatophyta were absent in 1989 and 2008 (Fig. 3).

In biogeographical terms, Cosmopolitan taxa ranked first each year, followed by Atlantic. Other interesting features were the low number of Mediterranean taxa and the increase in Indo-Pacific taxa from 2011 onwards (Fig. 4).

The dominant taxa, reaching a maximum yield of at least 5 kg m^{-2}, differed over the years. Table 3 shows the biomass values of these taxa each year, with the values shown in brackets when lower than 5 kg m^{-2}.

Considering the study period as a whole, the yearly biomass values (mean ± SD) for the whole basin ranged from 1.4±0.9 kg m^{-2} in 2014–2015 to 1.8±1.4 kg m^{-2} in 2017–2018 (Fig. 5a), see Suppl. materials 1–7: "Biomass data" for the original data used to perform this analysis. The site with the highest abundance by far was Station 1, with a pluriannual mean biomass of 2.5±1.2 kg m^{-2} (Fig. 5b). Concerning the season, there were no marked differences (R_{ANOSIM} = 0.017; p = 0.13), even though lower values were recorded in autumn than in the other seasons (Fig. 5c).

Species richness (S) and ecological diversity (H') increased over the years (Fig. 5d, g). The highest S was reached in 2016–17 when a total of 63 taxa (12.1±4.4 taxa per sample on average) were present and H' reached 1.2±0.4. Station 4 reached the high-

Table 2. Non-indigenous taxa recorded each year in the Mar Piccolo.

Species	1989	2008	2011	2012	2013	2014	2015	2016	2017	2018
Agardhiella subulata (C. Agardh) Kraft *et* M.J. Wynne	+							+	+	
Caulerpa cylindracea Sonder		+			+					
Codium fragile (Suringar) Hariot subsp. *fragile*			+		+	+	+	+	+	
Colpomenia peregrina Sauvageau				+	+	+	+	+	+	+
Dasysiphonia sp						+	+	+		
Grateloupia minima P.L. Crouan *et* H.M. Crouan							+		+	
Grateloupia turuturu Yamada			+	+	+	+	+	+	+	+
Hypnea cornuta (Kützing) J. Agardh		+	+	+	+	+	+	+	+	
Hypnea spinella (C. Agardh) Kützing	+	+				+				
Osmundea oederi (Gunnerus) G. Furnari		+							+	+
Polysiphonia morrowii Harvey				+		+	+	+	+	+
Solieria filiformis (Kützing) P.W. Gabrielson	+								+	
Undaria pinnatifida (Harvey) Suringar		+								
TOTAL 13	3	5	3	4	5	7	7	7	9	4

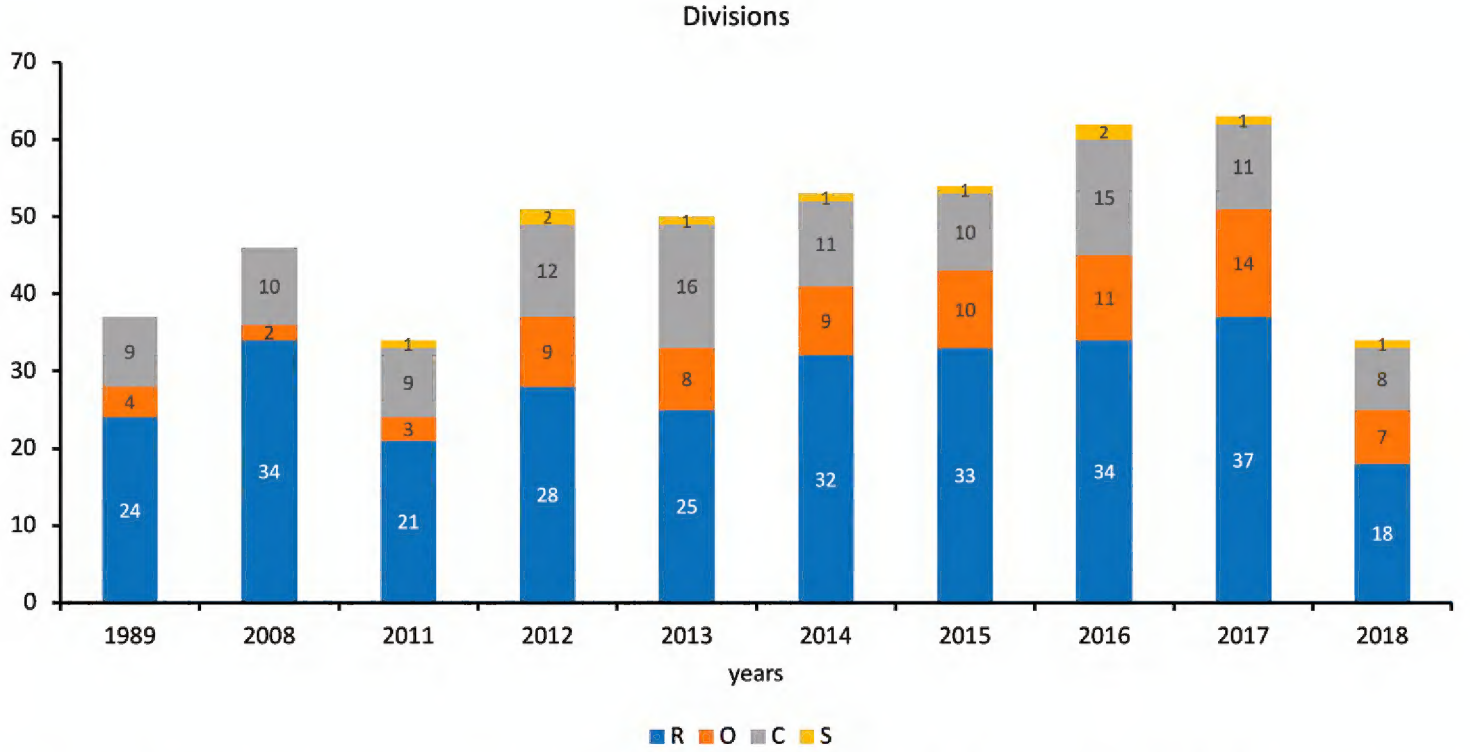

Figure 3. Trend in taxonomic divisions. 2018 data for winter only. In addition to the analysed years 2008 and 2011–2018, unpublished floristic data from 1989 have been included. R=Rhodophyta, O=Ochrophyta, C=Chlorophyta, S=Spermatophyta.

est S (12.4±4.5) and H'(1.2±0.4) (Fig. 5e, h). Higher S and H' values were observed in spring and winter than in autumn and, especially, in summer (Fig. 5f, i). J' values varied from 0.02 in autumn 2014 at Station 2 to 1 in autumn 2017 at Station 2.

Only small, marginally significant differences in the macrophyte community structure were apparent over the years (R_{ANOSIM} = 0.038; p = 0.03). The two-way crossed ANOSIM showed that there were no differences in community structure from one

Table 3. Maximum yearly yelds of dominant taxa (kg m^{-2}) in the Mar Piccolo of Taranto. In brackets values lower than 5 kg m^{-2}.

	2008	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018 (winter)**
Amphiroa beauvoisii	–	8.7	17.2	10.9	10.8	8.5	7.3	6.5	(0.93)
Chaetomorpha linum	29.9	10.8	11.3	(3.8)	5.4	(0.4)	6.1	(3.6)	(3.1)
Chondracanthus acicularis	(2.0)	(0.1)	(4.8)	6.8	(4.6)	(3.4)	(2.0)	11.6	(2.4)
Dictyota dichotoma var. *dichotoma*	(0.1)	(0.2)	(0.8)	(0.1)	(2.2)	6.5	10.3	11.4	–
Dictyota dichotoma var. *intricata*	(0.4)	6.3	(1.9)	9.5	6.9	(1.6)	(2.5)	9.3	10.0
Ellisolandia elongata	–	7.5	(1.4)	5.6	(2.6)	9.7	(1.4)	5.2	(0,1)
Hypnea cornuta	(2.3)	7.7	7.3	10.1	16.1	8.4	5.0	(4.1)	–
Spyridia filamentosa	(0.9)	(0.1)	(0.2)	(2.6)	(0.5)	(0.2)	(1.8)	7.1	(0.1)
Ulva intestinalis	–	–	(0.9)	–	(0.1)	–	5.1	(0.1)	(0.1)
Ulva laetevirens	(1.2)	9.7	(3.4)	(3.8)	(1.1)	(2.8)	(4.6)	(2.9)	(1.1)

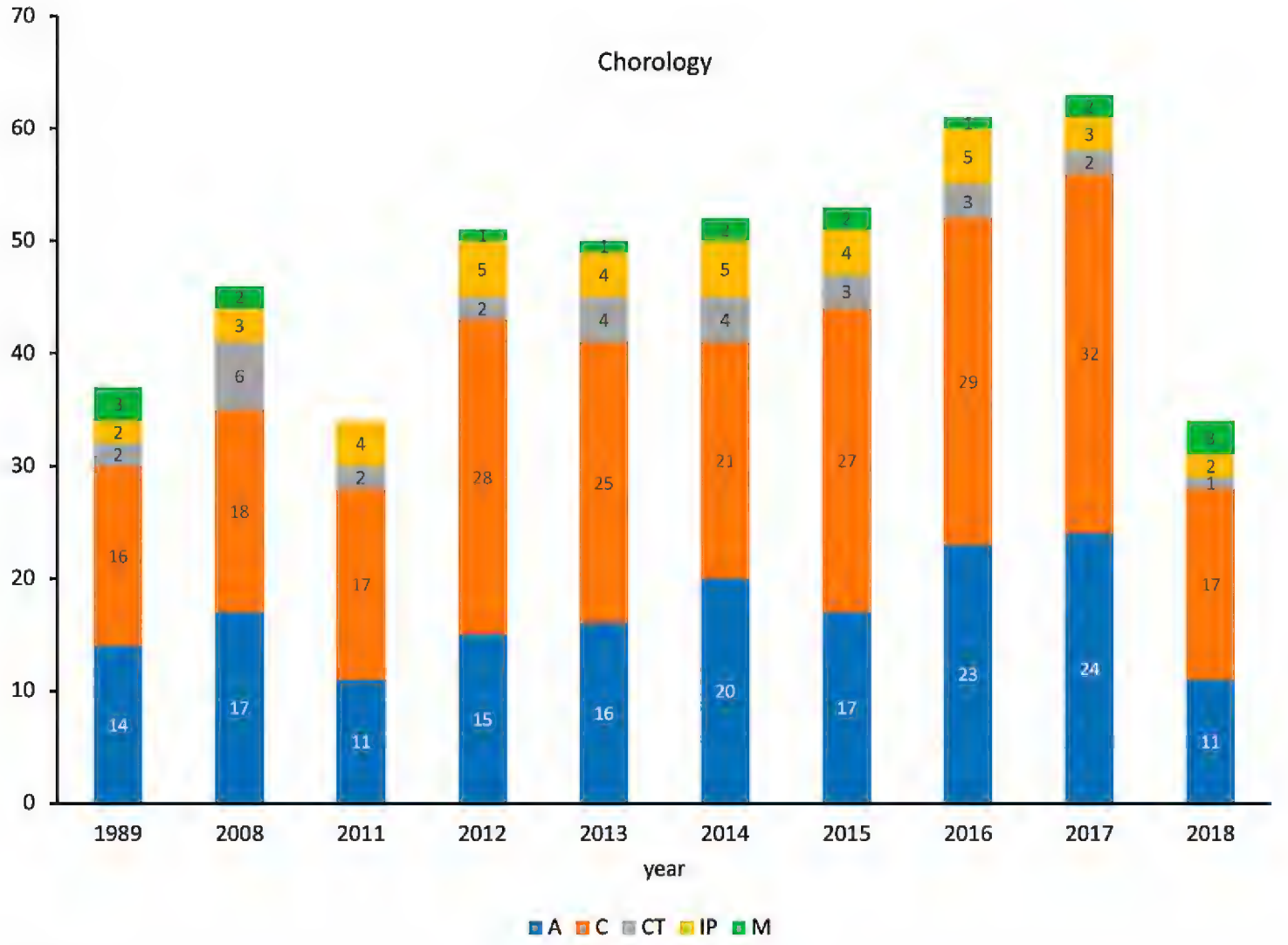

Figure 4. Chorological spectrum of taxa collected in the Mar Piccolo over the years. 2018 data for winter only. In addition to the analysed years 2008 and 2011–2018, unpublished floristic data from 1989 have been included. A=Atlantic, C=Cosmopolitan, CT=CircumTropical, IP=Indo-Pacific, M=Mediterranean.

year to the next, while there was variation between years distant from each other. For example, the pairwise test for 2011–2012 and 2017–2018 revealed significant variation in community structure (R = 0.363; p = 0.001), while the difference between

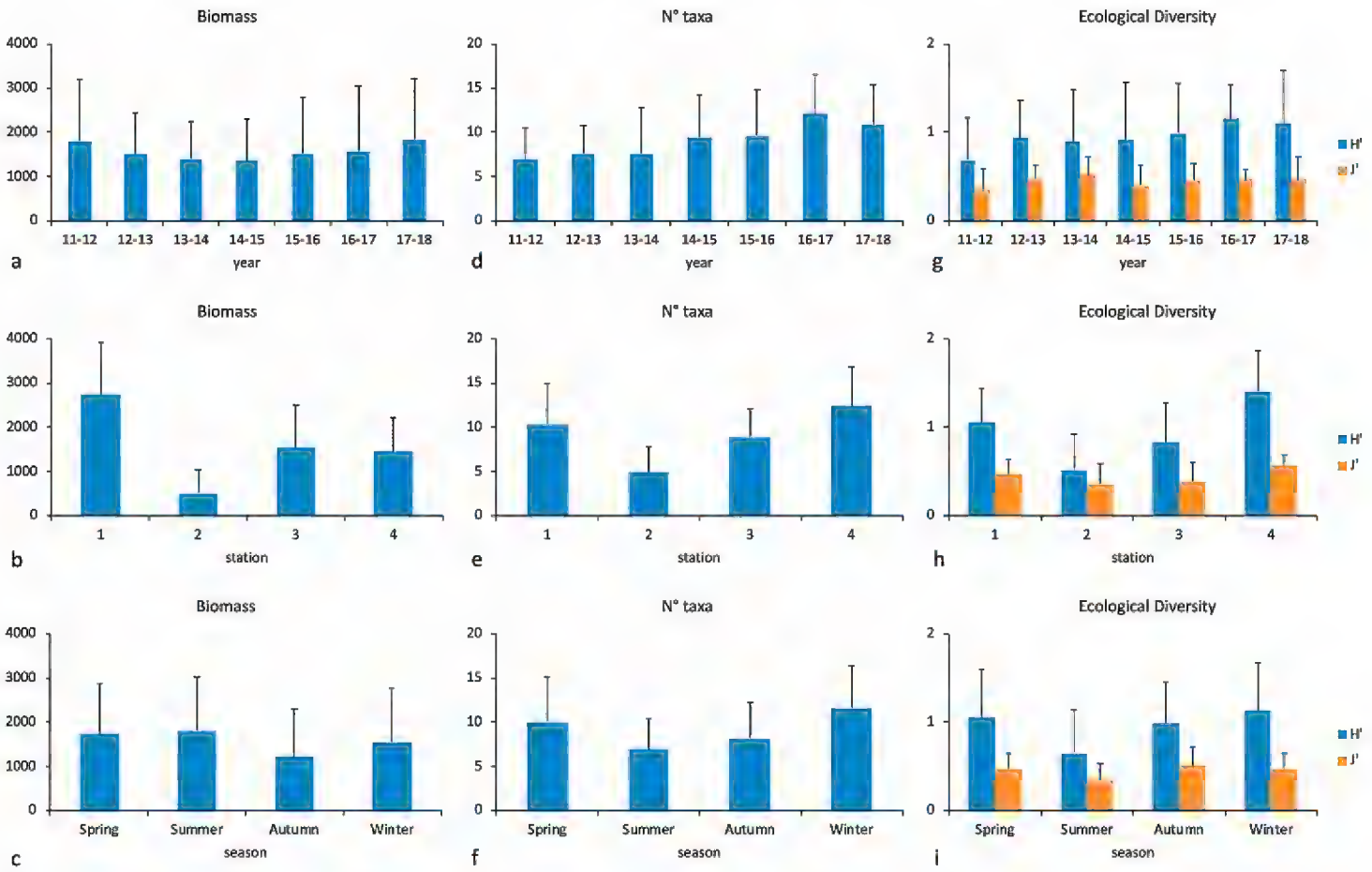

Figure 5. Mean values (±SD) calculated for the years of the study, sampling sites and seasons. **a, b, c** biomass (g m^{-2}) **d, e, f** species richness **g, h, i** ecological diversity.

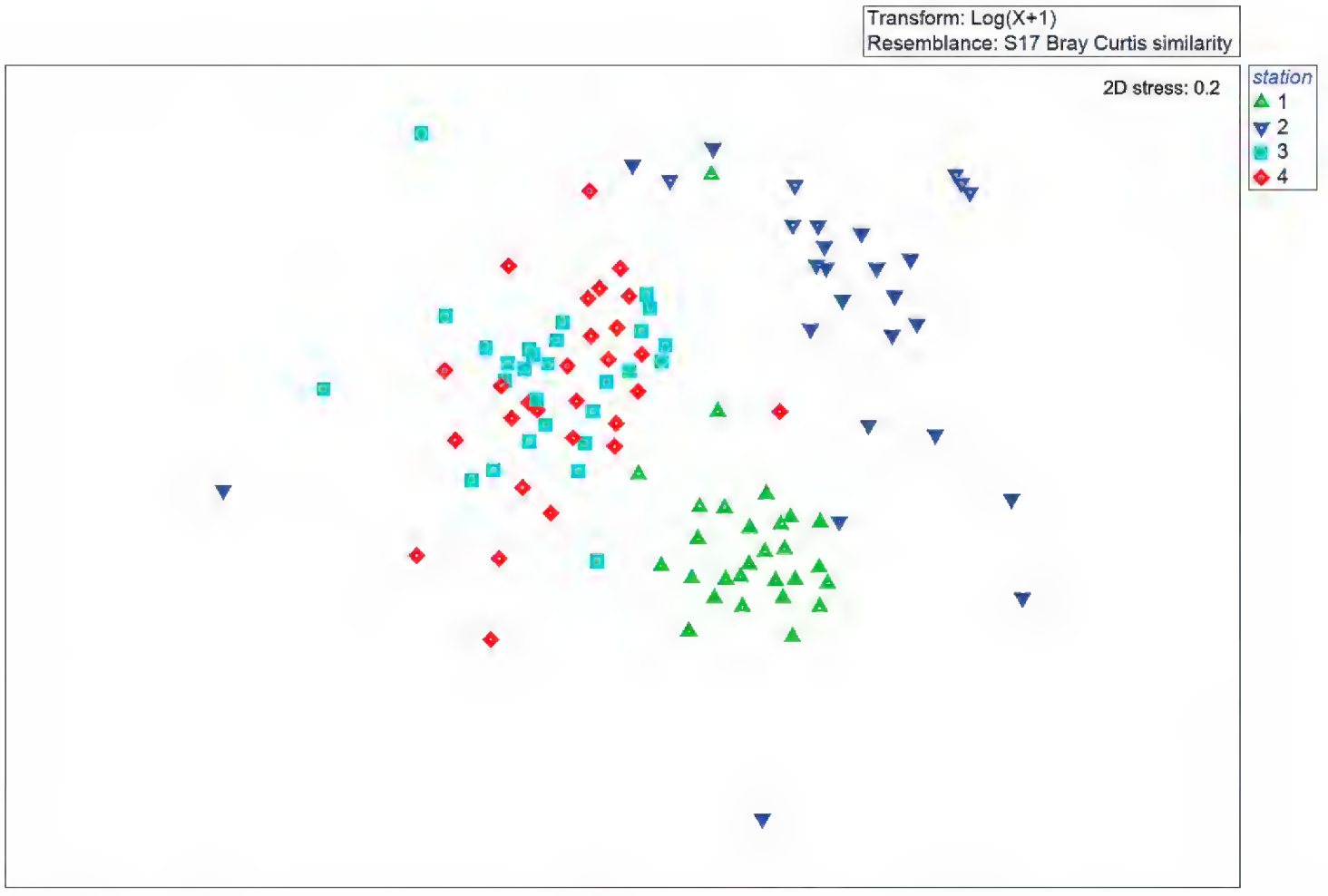

Figure 6. nMDS representation of the mean total biomass values reported for each sampling site in the period 2011–2018.

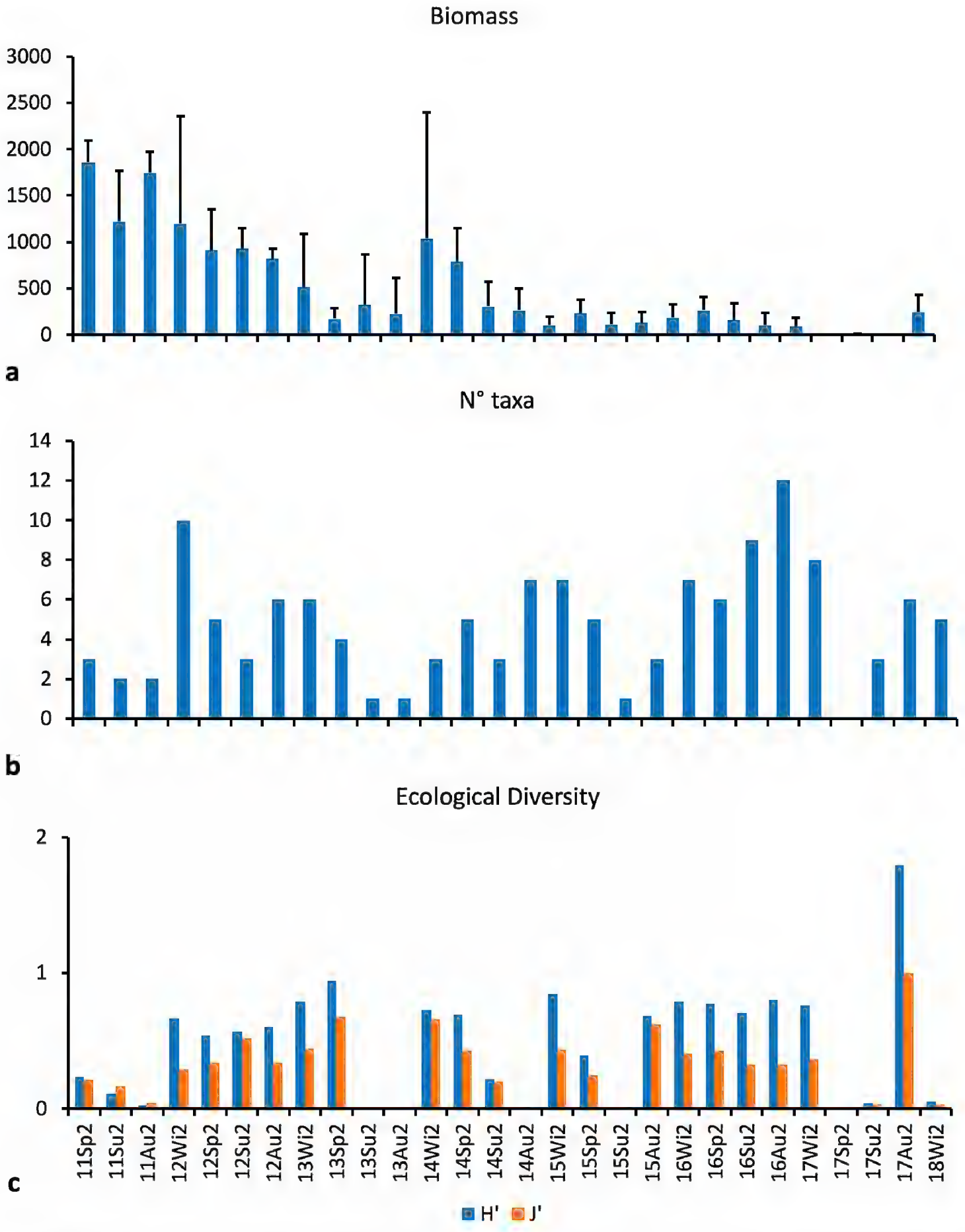

Figure 7. Mean values reported for Station 2 in the period 2011–2018. **a** biomass (g m^{-2}) **b** species richness **c** ecological diversity.

2016–2017 and 2017–2018 was not significant (R = 0.363; p = 0.3). Indeed, for immediately successive years, the R statistic was almost always negative.

The nMDS representation of the total biomass means per Station shows that the two stations in the Second Inlet were very close to each other and different from those

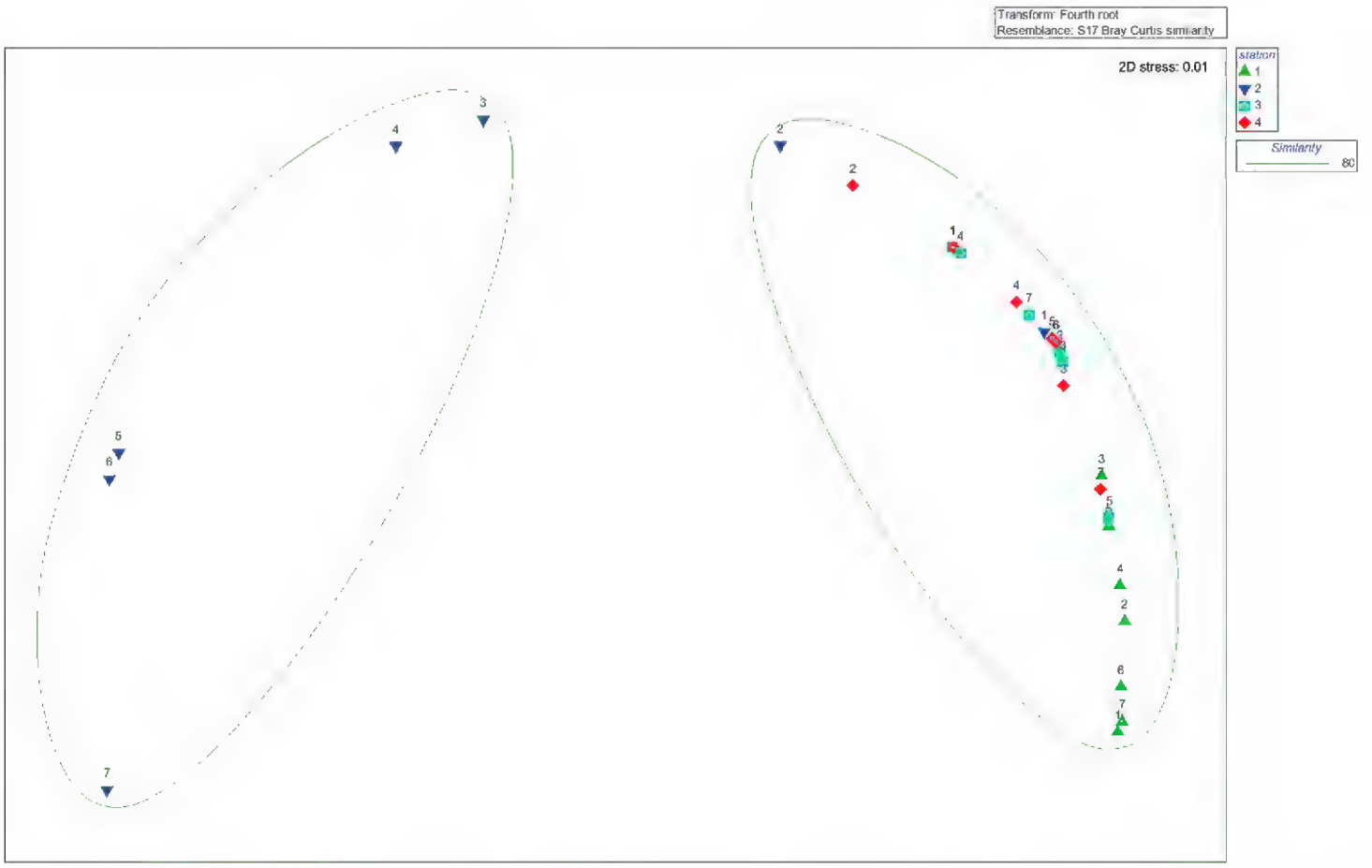

Figure 8. nMDS representation of the biomass means calculated over the period 2011–2018 at the four sampling sites. In the plot, the symbols indicate the station and the numbers refer to the sampling year (**1** 2011–2012 **2** 2012–2013 **3** 2013–2014 **4** 2014–2015 **5** 2015–2016 **6** 2016–2017 **7** 2017–2018).

in the First Inlet (Fig. 6). In addition, the analysis performed at the basin scale revealed a significant difference between the two inlets (R_{ANOSIM} = 0.47; p = 0.001).

Considering the analysis of each site, Station 2 was found to differ sharply from all the other stations (Fig. 7). Station 2 was found by the two-way SIMPER to have the highest dissimilarity in the pairwise comparisons with the other sites, higher than 97% in all cases. The ANOSIM confirmed this pattern (R_{ANOSIM} = 0.57; p = 0.001), all the pairwise tests showing high dissimilarity between stations, although Stations 3 and 4 in the Second Inlet were more similar to each other (R_{ANOSIM} = 0.169; p = 0.001). Station 2 had the lowest biomass and biodiversity (Fig. 5b, e, h). Regarding species richness and ecological diversity (Fig. 7b, c), Station 2 saw ups and downs over the years and a small recovery after 2016. Considering all the study years and sites, only 8.2% of the total biomass was recorded at Station 2, which saw a continuous fall in biomass values from 2011–2012 to 2017, when no macrophytes were collected in spring and negligible quantities were recorded in summer (4.9 ±8.4 g m^{-2}) and autumn (0.1 ±0.1 g m^{-2}). In winter 2018, higher values were measured again (Fig. 7a) when *Aglaothamnion tenuissimum* appeared, with relatively high biomass values (241 g m^{-2}).

Within the nMDS plot of the biomass means calculated over the years versus the sampling sites, in the first two years (2011–2012 and 2012–2013), Station 2 grouped with all the other sites, while in the other five years, sharp segregation was evident (Fig. 8).

Station 3 showed the highest abundance of NISs, both for the study period as a whole and year by year (Fig. 9b; Fig. 10). Starting from 2014–2015, Station 4 became more important (Fig. 10), reaching high biomass values, the highest NIS richness and ecological diversity (Fig. 9e, h). The biomass change was mainly accounted for by *Hypnea cornuta*, which reached 87% of total abundance in 2014–2015.

Discussion

LTER observations make it possible to detect the natural variability of ecological systems and the interaction between abiotic and biotic variables, as well as the effects on the environment of human activities (Kratz et al. 2003). Targeted studies have shown that seaweed species are strongly influenced by seawater temperature, salinity and other abiotic factors (Sousa-Dias and Melo 2008; Gallon et al. 2014).

In the Mar Piccolo, comparison of the situation dating back about thirty years and the results of the recent seven-year study show that qualitative and quantitative changes have occurred in the phytobenthic community and are ongoing. The statistical analysis showed that for the period 2011–2018, the structure of the phytobenthic community in the two basins of the Mar Piccolo differed and that a significant difference was also apparent between the sites investigated in the First Inlet, mainly due to the peculiar features of Station 2. At the same time, the results from the two stations of the Second Inlet were more similar even though with a significant difference in their community structure. Temporal variation was also observed, which was more evident when comparing distant years, while no significant differences were observed between successive years. Considering that the basin has been subject to a significant human pressure for centuries (especially from urban pollution and mussel farming) (Petrocelli et al. 2009) and that the last few years have seen the implementation of measures that have led to a change from eutrophic to oligotrophic conditions (Caroppo et al. 2011; Kralj et al. 2016), it may be assumed that most of these changes were of anthropogenic origin, beyond the natural alternation of macrophyte species (Ballesteros 1991).

Similar developments occurred over the 20 years from 1983 to 2003 in the Venice Lagoon, where a marked change in species composition was recorded (Sfriso and Curiel 2007). The disappearance of some species, generally considered as indicators of good environmental status, was attributed to the increase in pollution and eutrophication (Sfriso and Curiel 2007).

In the Orbetello Lagoon, a clear and progressive switchover of species occurred in the period 1983–2011, reflecting the worsening of ecological conditions in the lagoon, with the rise of *Chaetomorpha linum* (Lenzi et al. 2013). Over about 50 years (1953–2010), in the Nha Trang Bay (Vietnam, South China Sea), an increase in Chlorophyta, mainly Caulerpaceae, Cladophoraceae and Ulvaceae, was recorded (Titlyanov et al. 2015). Concurrently, Rhodomelaceae and Corallinaceae (Rhodophyta) and Sargassaceae and Dictyotaceae (Ochrophyta) decreased considerably. Chemical pollution

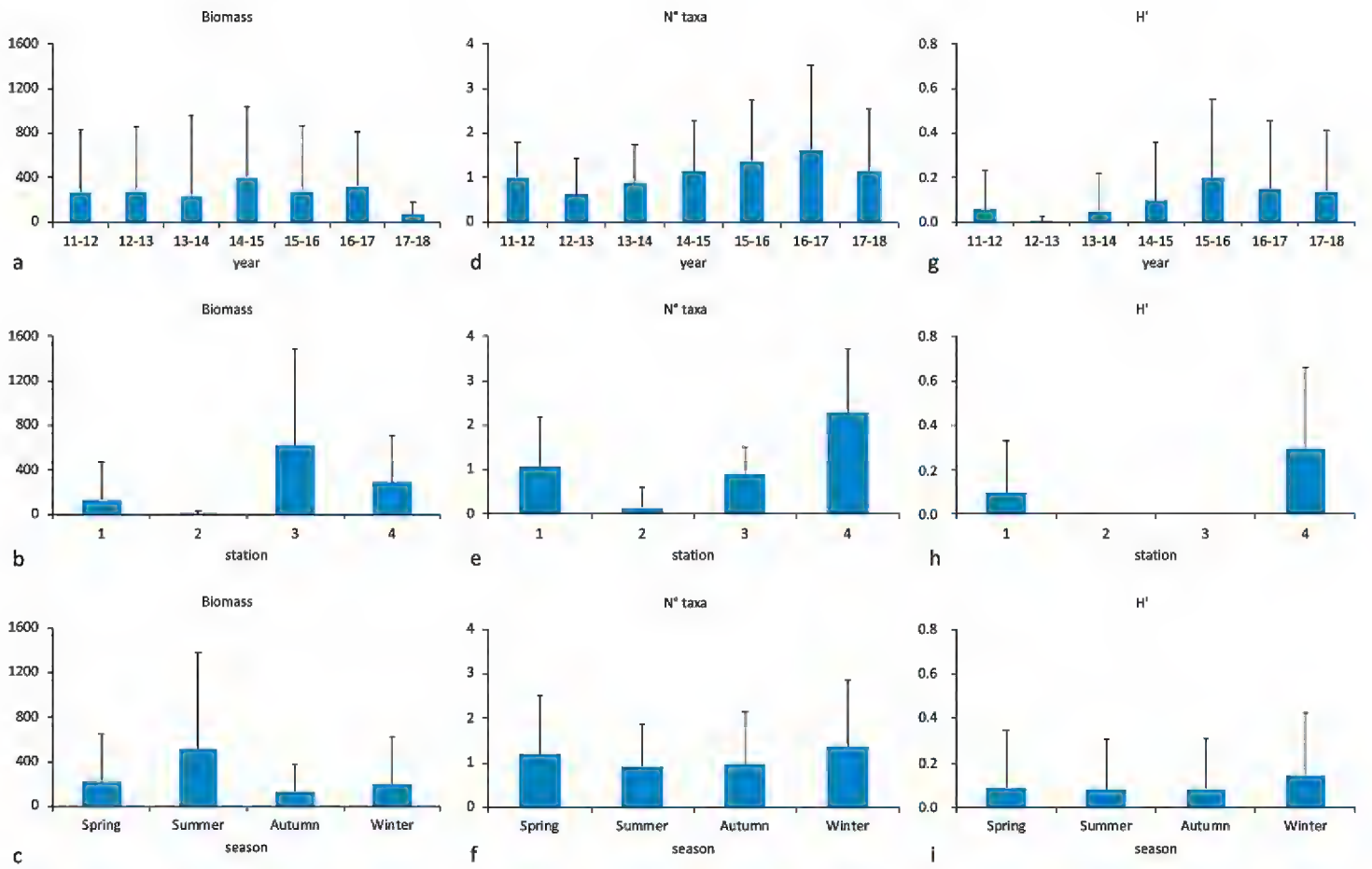

Figure 9. Mean numbers (±SD) of NISs in the period 2011–2018, sampling sites and seasons. **a, b, c** biomass (g m^{-2}) **d, e, f** species richness **g, h, i** ecological diversity.

of seawater due to the presence of inorganic and organic nitrogen and phosphorus compounds was held responsible (Titlyanov et al. 2015). Along the coast of the Gullmar Fjord (W Sweden), data on macroalgal vegetation, dating back to 1941, allowed a comparison with the situation over 40 years later, in 1998. Changes in species composition and depth distribution were observed, including a decrease in coarse filamentous and perennial species and an increase in small filamentous and ephemeral species, which were ascribed to the increased nutrient load (Eriksson et al. 2002). Since the 1950s, macroalgal communities in the Hardangerfjord (Norway) have seen a considerable increase in the number of species, mainly southern species with warm-water affinity and NISs, most probably due to the increase in seawater temperature (Husa et al. 2014). The increase in Rhodophyta was noteworthy, attributable to the installation of a hydro-electrical power plant which caused a change in salinity in the fjord (Husa et al. 2014). In a ten-year study carried out in Tancada Lagoon (NE Spain), the reduction in rooted macrophyte coverage was accompanied by an increase in unattached seaweeds. Amongst the latter, *Chaetomorpha linum* was the dominant species in autumn and winter, when it flourished with high concentrations of dissolved ammonium salts (Menéndez and Comín 2000).

Considering the floristic aspect, most of the species detected in the Mar Piccolo have also been reported in similar Mediterranean environments such as Greek and Cypriot TWSs (Christia et al. 2011) and French lagoons (Le Fur et al. 2018). Twenty-nine of them are also frequently found in euhaline Atlanto-Mediterranean coastal la-

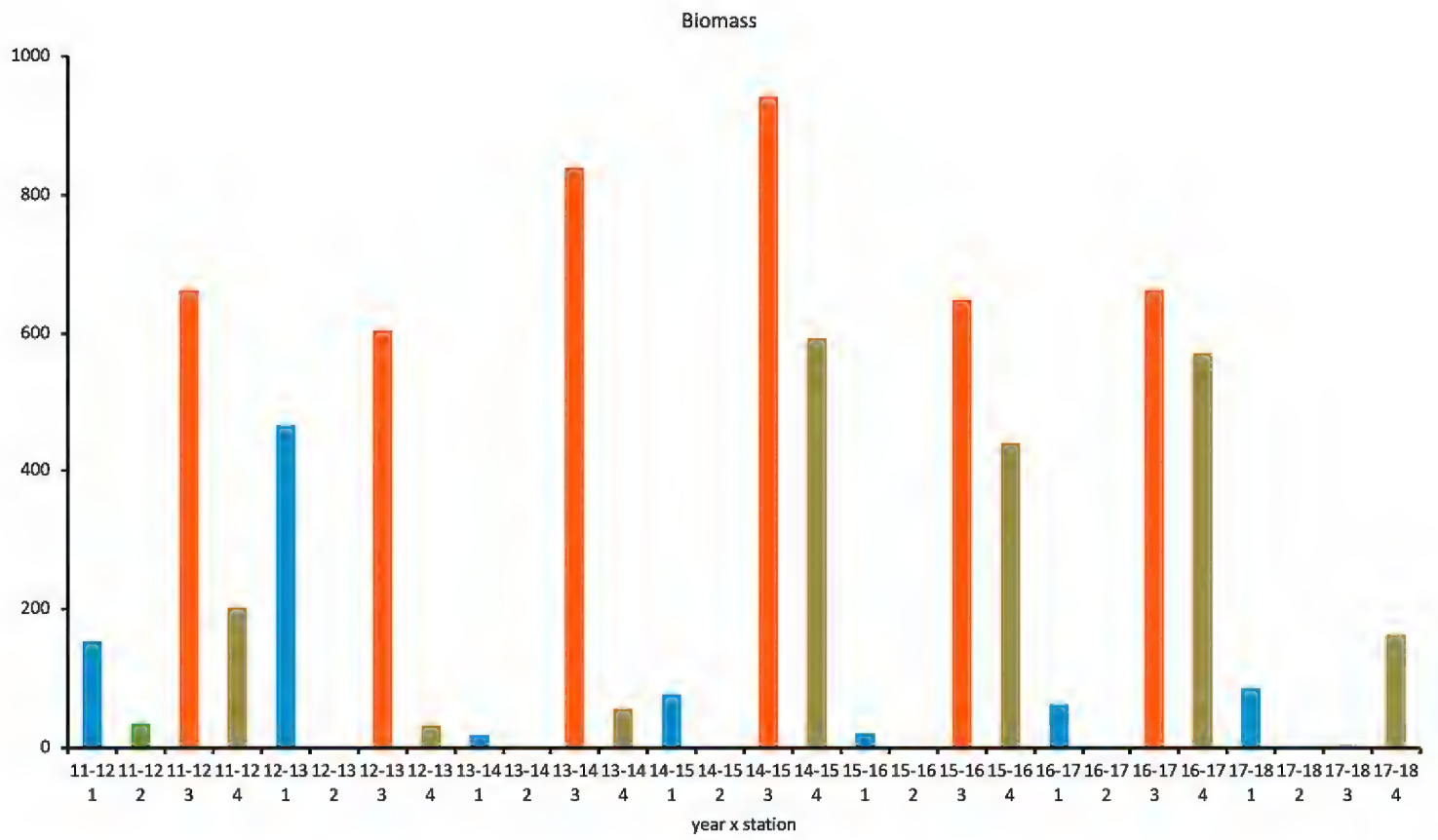

Figure 10. NIS biomass values (g m^{-2}) calculated for each year of the study at the four sampling sites.

goons (Pérez-Ruzafa et al. 2011b). These are mainly euriecious species, less sensitive to the continuous changes in the chemical and physical features of these environments (e.g. *Gracilaria bursa-pastoris* and *Ulva intestinalis*), but also NISs, most likely introduced through the importation of molluscs (e.g. *Colpomenia peregrina* and *Grateloupia turuturu*) (Cecere et al. 2011; Petrocelli et al. 2013). The predominance of Rhodophyta is seen to be a typical pattern in most TWSs (Falace et al. 2009; Ramdani et al. 2015; Le Fur et al. 2018), except for environments characterised by noticeable surges in nutrient concentrations and salinity, where Chlorophyta are dominant (Piñón-Gimate et al. 2008; Pellizzari et al. 2009). However, the most evident development in the last few years in the Mar Piccolo is the increase in Ochrophyta. This was probably due to the improvement of seawater quality following the closure of sewage pipes in the First Inlet (Alabiso et al. 2005; Kralj et al. 2016). Indeed, Ochrophyta are acknowledged to be the most sensitive seaweed species to urban pollution, which strongly affects their reproduction and physiology (Scherner et al. 2012).

Likewise, the chorological spectrum of Mar Piccolo phytobenthos has seen considerable fluctuation. The present situation is quite different from that of Italian marine flora in general, where the Atlantic and Mediterranean elements are prevalent, while the Indo-Pacific and Circum-Tropical elements are practically negligible (Furnari et al. 2010). Specifically, the increase in the Indo-Pacific element in the Mar Piccolo is most probably due to the introduction of NISs. In the Thau Lagoon (France), about 30% of seaweed taxa (i.e. 58 out of 179 taxa) are NISs native to the western Pacific. They are so numerous that the lagoon is considered "a Japanese biological island in the Mediterranean Sea" (Boudouresque et al. 2011). Their presence is considered a result

of the intensive importation of Japanese oysters for aquaculture purposes in the early 1970s (Boudouresque et al. 2011). In the Venice Lagoon, 20 NISs are of Pacific origin (Marchini et al. 2015). Here, the rise in NISs is attributed to the development of commercial activities such as ship traffic and mollusc importation (Sfriso and Curiel 2007). Similarly, the continuous increase in NISs in the Mar Piccolo could be ascribed to the massive importation of shellfish as a result of the serious crisis affecting Taranto mussel farming since 2011 (Cecere et al 2016b).

The total of 129 taxa recorded in the Mar Piccolo in the 2011–2018 period is probably an underestimate, since only four coastal stations were seasonally sampled from a total surface area of about 21 km^2. However, it was still comparable with that of other Mediterranean TWSs, considering that some of these environments have larger surface areas and probably offer a higher range of environmental conditions (Table 4).

Quantitative dominance showed the same fluctuations amongst both years and seasons. In 1989, Gracilariaceae and Solieriaceae had the highest standing crop throughout the basin (Cecere et al. 1992). The period 2011–2018 saw continuous changes in dominance amongst species (Table 3), unlike 2008, when only *Chaetomorpha linum* reached noticeable biomass values. The increase in geniculate Corallinaceae (e.g. *Amphiroa beauvoisii* and *Ellisolandia elongata*) was most probably linked to the decrease in nutrient load, as already observed in Spanish waters (Pinedo et al. 2013). A decrease of about one order of magnitude in the standing crop of submerged macrophytes was observed over about twenty years in the Lagoon of Venice, which also saw a marked dominance of seagrasses over seaweeds, with higher standing crops for *Cymodocea nodosa* (Sfriso and Facca 2007).

Considering differences in spatial patterns, it is well known that coastal lagoons are "a mosaic of assemblages", mainly depending on abiotic factors (Pérez-Ruzafa et al. 2008). In the Mar Piccolo, this was already highlighted in 1989, when noticeable spatial differences were recorded mainly due to a marked variability of nutrient concentrations. Specifically, Zone B, the nearest to Station 2, had the lowest biomass and biodiversity (Cecere et al. 1992). Therefore, the present study not only confirms previous observations, but, via multivariate analysis, sheds light on the remarkable change that has happened at this site since 2013. Specifically, the two-way SIMPER (years vs. sites) indicates that, at Station 2, the change was due to a huge decrease in biomass and not an apparent change in community composition. As already indicated by the univariate analysis, in 2016, an increase in species richness and diversity was registered here. However, *Chaetomorpha linum* and *Cymodocea nodosa,* which are considered to be structuring taxa, alone represented 80% of the total biomass, the others being mainly epiphytic species on *C. nodosa*. Of the latter, *Aglaothamnion tenuissimum,* commonly reported in epiphytic habits, developed a considerable quantity of pleustophytic biomass – uncommon in this species – in winter 2018. The area surrounding Station 2 is characterised by the presence of a small river discharging fresh water drained from surrounding farmland. The massive biomass development of *A. tenuissimum* is thus most probably due to an abnormal increase in nutrients, as already observed for epiphytes of *Posidonia oceanica* (Linnaeus) Delile (Prado et al. 2008), but it deserves to

Table 4. Number of seaweed and phanerogams taxa in some Mediterranean TWS. S=number of seaweeds; P=number of phanerogams.

TWS	S	P	Surface	Reference
Mar Piccolo	126	2	21 km^2	this study
Venice Lagoon	296	5	432 km^2	Sfriso et al. 2009
Thau Lagoon (France)	179	2	75 km^2	Boudouresque et al. 2010
Nadoor Lagoon (Morocco)	110	2	114 km^2	Ramdani et al. 2015
Stagnone of Marsala	108	4	20 km^2	Calvo et al. 2009
Mar Menor (Spain)	69	2	135 km^2	Pérez-Ruzafa et al 2008
Orbetello Lagoon	68	3	25 km^2	Lenzi et al. 2009
Marano and Grado Lagoon	41	4	160 km^2	Falace et al. 2009
Acquatina Lake	38	2	0.45 km^2	Petrocelli and Cecere 2010
Ganzirri Lake	32	3	0.34 km^2	Serio et al. 2009
Lesina Lagoon	36	2	51.36 km^2	D'Adamo et al 2009, E. Cecere and A. Petrocelli unpublished data
Caprolace Lagoon	28	3	2.26 km^2	Signorini et al. 2009a
Faro Lake	28	3	0.26 km^2	Serio et al. 2009
Fogliano Lagoon	10	2	3.95 km^2	Signorini et al. 2009b

be investigated further. The highest abundance of the NIS *Hypnea cornuta* at Station 3 may, in contrast, be ascribable to the presence of natural hard substrata, unique in the basin. Indeed, this species is characterised by organs of vegetative propagation (i.e. stellate propagules, thickened unattached fragments, basal parts) which also serve as over-wintering organs. In late autumn-early winter, the thalli decay but those organs survive the adverse season (Cecere et al. 2004, 2016a). The natural cavities in the rock at Station 3 probably represent a shelter supporting the survival of a higher number of propagules.

Considering a longer time period of about one century, complemented with information on changes in both the demography of Taranto, based on decade-long census data (https://tinyurl.com/yacl8t6t) and the forcing factors affecting the Mar Piccolo, changes in phytobenthos can speculatively be related to human pressure.

In the 1920s, Taranto had fewer than 100,000 inhabitants (https://tinyurl.com/yacl8t6t) and the only anthropogenic factors affecting the Mar Piccolo were mussel breeding and the shipyard of the Italian Royal Navy, established in 1889 (Nistri 1988). In that period, the waters of the Mar Piccolo seemed to be in good condition, hosting seaweed species commonly recorded in open and clean seawaters (Pierpaoli 1923), together with luxuriant meadows of the phanerogam *Cymodocea nodosa* (Pierpaoli 1923).

In the 1940s, a change was already observed, linked to the presence of the First Squadron of the Italian Royal Navy fleet during the Second World War (Pierpaoli 1959). Considerable quantities of both fuel and the residues of smoke bombs were observable in the areas where the ships were moored and where the anti-aircraft defences were placed. This probably adversely affected seawater quality and seaweed assemblages, with Ulvales growing on the docks and a noticeably lower number of Ochrophyta and Rhodophyta species (Pierpaoli 1959).

The period from the 1950s to the 1970s saw a considerable increase in the population of Taranto (https://tinyurl.com/yacl8t6t), linked to the development of new economic activities, chief amongst which was the ITALSIDER steelworks (now Arcelor-Mittal Italy). New districts were built and the town expanded (https://tinyurl.com/ycm2y68w, Nistri 1988). Along the shores of the Mar Piccolo, shipyard activities decreased (https://tinyurl.com/ya885vun) and the Italian Air Force training school was built (https://tinyurl.com/y8wpbgpu). Unfortunately, no information on the phytobenthos of the basin in that period is available, except for some notes in publications of general interest (Parenzan 1984).

At the beginning of the 1980s, the population of Taranto peaked along with the industrial activities (https://tinyurl.com/yacl8t6t). The demographic and economic boom resulted in the creation of 14 sewage outlets, both urban and military, which discharged untreated effluent into the Mar Piccolo until the late 1990s. The most immediate consequences of these factors were the eutrophication of the basin's waters and an increase in turbidity due to suspended particulate matter (Pastore 1993). Subsequent phytobenthos studies highlighted considerable changes with respect to Pierpaoli's data (Cecere et al. 1991a), with the following becoming more evident: excessive growth of nitrophilous species; the unattached status of most of the seaweeds; the loss of sexual reproduction and the onset of vegetative reproduction; the disappearance of the less tolerant species, amongst which brown seaweeds and in particular those of the *Cystoseira* genus; and the thinning out of the *Cymodocea nodosa* meadow (Cecere et al. 1988, 1989, 1991b; Saracino and Cecere 1991; Cecere et al. 1992). This situation was in line with observations in other coastal environments, where urban, agricultural and industrial sewage caused eutrophication, with a high level of inorganic nutrients and turbidity (De Casabianca et al. 1997). It is well known that untreated sewage discharged into coastal basins is the cause of severe modifications in benthic communities, due to increases in nutrients, toxic substances, turbidity and silt. Generally, in these conditions, filamentous and sheet-like algae prevail over thick algae, due to their higher uptake efficiency and faster growth (Díez et al. 2013). Specifically, in these environments, macrophytes represent an important structuring component. When nutrient levels rise, one of the first feedbacks is the shift of phytobenthic communities from rooted macrophytes and perennial seaweed species towards fast-growing and ephemeral macroalgae (De Casabianca et al. 1997; Obrador and Pretus 2010,; Scherner et al. 2013). In the Thau Lagoon, it was observed that the increase in eutrophication, mainly due to mussel farming activities, caused profound changes in macrophytobenthic communities. Pure eelgrass meadows (i.e. *Zostera marina* and *Nanozostera noltii*) were progressively replaced by nitrophilous communities of *Ulva rigida* and *Gracilaria bursa-pastoris* (De Casabianca et al. 1997). A quantitative study carried out along the Atlantic coast of Brazil proved that, in highly urbanised areas, calcareous algae coverage fell, most probably due to high concentrations of nutrients, which inhibited calcium carbonate synthesis. Another effect was the reduction in biodiversity and changes in macroalgal population structure. Indeed, Ochrophyta considerably decreased and Chlorophyta increased (Scherner et al. 2013).

The beginning of the new century saw the start of a new era; the population in Taranto began to decrease (https://tinyurl.com/yacl8t6t) and nine urban sewage outlets were closed (Caroppo et al. 2011). The physico-chemical features of the seawaters changed again, becoming less eutrophic and in some areas recovering their transparency (Alabiso et al. 2005; Kralj et al. 2016). A "re-oligotrophication trajectory" was thus triggered, with relatively rapid results (Le Fur et al. 2019). In addition, unlike other studies, the availability of historical data confirmed that a real recovery was in progress (Le Fur et al. 2019). Indeed, seaweed species, typical of clear seawater recorded by Pierpaoli (1923), reappeared (Petrocelli et al. 2014). In addition, many species recovered their capacity for sexual reproduction (Petrocelli et al. 2009), the *Cymodocea nodosa* meadow increased in density and plants began to produce flowers and fruits. A luxuriant meadow of another phanerogam, *Ruppia cirrhosa*, never previously recorded, was observed, with flowers and fruits (Petrocelli et al. 2009; Cecere et al. 2010). Similar occurrences have been observed in other transitional systems. A 22-year study of the recovery of seaweed populations after the installation of a sewerage system was carried out in the Abra de Bilbao (North Spain), a semi-enclosed bay, which for a long time had received urban and industrial effluent through a river (Díez et al. 2009, 2013). The increased algal cover at once degraded sites, the rise in the number of species concomitant with the improvement in seawater quality and the increase in similarity amongst stations were the most noteworthy results (Díez et al. 2009). An evaluation of long-term changes in macroalgal populations present around the coast of Catalonia (Spain, Mediterranean Sea) was performed over 17 years, in order to assess the effects of a sewerage system. An evident switch towards a decrease in both the number and abundance of nitrophilous Chlorophyta (e.g. *Cladophora* spp. and *Ulva* spp.) in favour of *Ellisolandia elongata* (as *Corallina elongata*), a stress-tolerant species, was recorded (Pinedo et al. 2013). The Saronikos Gulf (Greece, Aegean Sea) underwent a similar process, the heavily polluted conditions caused by rapid urbanisation from the mid-1950s to the mid-1990s giving way to oligotrophic seawaters by 1998, thanks to the installation of a sewerage system (Tsiamis et al. 2013). In the initial period, macroalgal communities were characterised by the dominance of nitrophilous Chlorophyta species such as *Cladophora* spp. and *Ulva* spp. Following the installation of the wastewater treatment system, seaweed communities became more biodiverse and Rhodophyta species, mainly Ceramiales, increased (Tsiamis et al. 2013). In Australia, near two decommissioned sewage outfalls, a decrease in species generally flourishing in eutrophic waters (e.g. *Ulva lactuca* and *Corallina officinalis*) was recorded, while Ochrophyta (e.g. *Colpomenia sinuosa* and *Petalonia fascia*) and Rhodophyta species (e.g. *Pterocladiella capillacea,* encrusting Corallinaceae) increased in abundance (Archambault et al. 2001).

Conclusions

The Italian LTER network combines a number of marine, terrestrial and freshwater ecosystems where ecological data have been collected for several decades. By means of interdisciplinary activities and cross-ecosystem research, it aims to make historic

datasets, in series of up to a century long, available for ecological research and preserve them for future generations (Bertoni et al. 2012).

In the Mar Piccolo, LTER studies enabled the analysis of historic qualitative and quantitative data and made it possible to draw up a history of its phytobenthos over about one century. On the basis of multidisciplinary observations of the whole basin conducted over many years (Petrocelli et al. unpublished data), it may be assumed that human activities, directly or indirectly, were the main cause of the changes. However, it is important to highlight that the basin showed high resilience. Indeed, following the removal of most of the urban wastewater discharges, which were the main cause of organic pollution, the basin was able to return to better conditions in just a decade. Therefore, it is realistic to hope that the Mar Piccolo could once again become the paradisiacal place described in ancient times, if only individual users adopt good habits. Indeed, strong pressures on the basin also arise from the careless dropping of litter of all types, such as mollusc shells, plastics and engine oil from fishing boats. In any case, long-term studies are ongoing, so it will be possible to monitor the changes year by year.

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Supplementary material 1

Biomass data in 2011–2012

Authors: Antonella Petrocelli, Ester Cecere, Fernando Rubino

Data type: measurement

Explanation note: Mean biomass values (g m-2), standard deviation and total number of taxa measured in each station and in each season in 2011–2012.

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Link: https://doi.org/10.3897/natureconservation.34.30055.suppl1

Supplementary material 2

Biomass data in 2012–2013

Authors: Antonella Petrocelli, Ester Cecere, Fernando Rubino

Data type: measurement

Explanation note: Mean biomass values (g m-2), standard deviation and total number of taxa measured in each station and in each season in 2012–2013.

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Link: https://doi.org/10.3897/natureconservation.34.30055.suppl2

Supplementary material 3

Biomass data in 2013–2014

Authors: Antonella Petrocelli, Ester Cecere, Fernando Rubino

Data type: measurement

Explanation note: Mean biomass values (g m-2), standard deviation and total number of taxa measured in each station and in each season in 2013–2014.

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Link: https://doi.org/10.3897/natureconservation.34.30055.suppl3

Supplementary material 4

Biomass data in 2014–2015

Authors: Antonella Petrocelli, Ester Cecere, Fernando Rubino

Data type: measurement

Explanation note: Mean biomass values (g m-2), standard deviation and total number of taxa measured in each station and in each season in 2014–2015.

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Link: https://doi.org/10.3897/natureconservation.34.30055.suppl4

Supplementary material 5

Biomass data in 2015–2016

Authors: Antonella Petrocelli, Ester Cecere, Fernando Rubino

Data type: measurement

Explanation note: Mean biomass values (g m-2), standard deviation and total number of taxa measured in each station and in each season in 2015–2016.

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Supplementary material 6

Biomass data in 2016–2017

Authors: Antonella Petrocelli, Ester Cecere, Fernando Rubino

Data type: measurement

Explanation note: Mean biomass values (g m-2), standard deviation and total number of taxa measured in each station and in each season in 2016–2017.

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Supplementary material 7

Biomass data in 2017–2018

Authors: Antonella Petrocelli, Ester Cecere, Fernando Rubino

Data type: measurement

Explanation note: Mean biomass values (g m-2), standard deviation and total number of taxa measured in each station and in each season in 2017–2018.

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